Biomonitoring of the Derwent River

at Derwent Bridge: 1995 - 2003

Report to Department of Primary Industry, Water and Energy

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Freshwater Systems

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Introduction

This report describes the cumulative results to March 2003 of the biological monitoring conducted in the Derwent River at Derwent Bridge prior to and following the commencement of discharge of the wastewater treatment plant (WWTP) constructed to treat waste waters from developments at Cynthia Bay at Lake St Clair. Monitoring of the biota of the ambient waters receiving discharge from the WWTP is required under the conditions of license to operate by the Department of Primary Industry Water and Environment (DPIWE). As part of those conditions, monitoring at two sites was required during a period prior to commencement of WWTP discharge (1995) and thereafter (1996, 1997, 1998, 2000). Sampling was conducted three times per year in 1995-1997 (once each in summer, autumn and winter), was decreased to twice a year in 1998 (once in autumn and winter). In 1998, the frequency of sampling was further decreased to once every second year, in autumn, with sampling conducted in April 2000 and March 2003.

Pre-discharge biological monitoring commenced in early 1995. Commencement of WWTP discharge commenced in August 1996. There are therefore, three sampling events conducted prior to discharge. These samples represent the ‘pre-discharge’ conditions, while all samples after late 1996 represent the ‘post-discharge’ conditions.

The two sampling sites are situated 100 m upstream and downstream of the discharge point, located approx. 1 km downstream of Derwent Bridge. Monitoring consists of regular quantitative sampling of benthic (bottom dwelling) macroinvertebrates (principally aquatic insects and crustaceans) and algae in the Derwent River.

The discharge point is immediately upstream of Lake King William. At the point of proposed discharge, the Derwent River consists of a sequence of long runs interspersed with occasional deep pools. The runs are characterised by a gravel and small cobble substrate, grading to fine gravel and sand on the channel margins. Flows are partially controlled by HEC gates downstream of Lake St Clair, but the river hydrology is essentially natural, reflecting the discharge from Lake St Clair, whose storage characteristics have not been substantially altered by HEC operations and whose catchment is essentially pristine. The reach in which the WWTP discharge occurs, is, however, frequently inundated by Lake King William waters when that lake is within 1 - 2 m of full supply level (FSL). Prolonged periods of inundation lead to a decrease in the abundance and diversity of macroinvertebrates at both sites, to a similar extent (see Davies and Cook 1998a, 1998b, 2000).

Sites and Methods

Sites

Two study sites were selected as follows:

Site	Location	Easting	Northing
Upstream	100 m upstream of proposed discharge	436500	5334500
Downstream	100 m downstream of proposed discharge	436500	5334200

Sample sites were typified by a fine gravel to small cobble substrate.

Sampling

Sampling methods have been consistent throughout the study. Sampling for benthic macroinvertebrates was performed on three occasions prior to commencement of WWTP discharge - 25/1/1995, 27/2/1995 and 10/5/1995, and was then discontinued. Once WWTP discharge had commenced, the monitoring program was re-instated in December 1996. 'Post-discharge' sampling was thus conducted on 3/12/96; 27/2, 9/5 and 23/7/1997; 4/2 and 5/5/1998; 6/4/2000 and 17/3/2003.

Sampling was performed at randomly selected locations, mid-channel, on the dominant mixed gravel - small cobble substrate. Fifteen samples were collected at each site by hand disturbance and washing of the substrate to a maximum depth of 10 cm in a 30 x 30 cm quadrat immediately upstream of a standard Surber sampler fitted with a 500 µm mesh net.

Algal sampling was conducted using a scouring sampler, as described by Davies and Gee (1993). Algal biomass is assessed by extracting scoured algal material from a standard surface area and measuring the yield of chlorophyll-a. On each sampling occasion, 15 individual scourer samples were collected at random locations at each site from the upper surface of approx. 5 - 10 cm diameter cobbles with the scourer sampler.

Sample processing

Sampling processing methods have been consistent throughout the study. Surber samples were pooled and preserved with 10% formalin. They were later processed by washing on a 500 µm mesh sieve and subsampling using a 100 cell box sub-sampler, as described by Marchant (1991). A 20% sub-sample was taken from each pooled sample by selecting 20 of the 100 cells in a systematic, stratified random manner.

All sub-samples were then hand-picked and macroinvertebrates counted and identified to family level (with the exception of oligochaetes, nematodes and mites).

Each algal sample was stored on ice in the dark prior to extraction and analysis for chlorophyll-a content by a modified acetone extraction-spectrophotometric method (APHA 1993).

Data analysis

The total abundance and number of taxa were thus calculated for each pooled sample, as well as the abundance of each taxon. Bray Curtis Similarity values were calculated for each of the seven upstream-downstream pairs, to assess changes in community composition. In addition, a multidimensional scaling (MDS) ordination was conducted on the Bray Curtis Similarity matrix for all samples using the PRIMER software package (Carr 1996).

Differences between upstream and downstream sites in abundance and number of taxa were examined using analysis of variance, after log(x+1) transformation of the data. Two factors were used in the ANOVA's - site (upstream vs. downstream) and time (pre- vs. post-commencement of discharge i.e. 1995 vs. 12/1996-3/2003). The statistical significance of both factors, as well as factor interaction (site x time) were examined.

Results

Results of all sampling are shown in Table 1. All sites have consistently supported a diverse fauna, with around 20 taxa. On all occasions, they were numerically dominated by oligochaetes (worms) and the larval forms of chironomids (midges), leptophlebiid mayflies and leptocerid caddisflies. This fauna continues to be typical of reasonably clean Tasmanian stream sites with low to moderate levels of pollution.

Total macroinvertebrate abundances in March 2003 were markedly lower at both sites than in previous years. Diversity, as measured by the number of taxa, was consistent between sites and with samples from previous years. The fact that these were all taxa that have been characteristic of these sites since 1995, that these differences were consistent across sites, and that total diversity had not changed nor was different indicate that there has been no significant impact on the fauna from the discharge, and that both sites have experienced natural decreases in faunal density.

Chlorophyll-a levels were low in March 2003, consistent with previous years (all < 30 mg/m^2) and again consistent with an oligotrophic stream site (i.e. no evidence of nutrient enrichment).

For the year 2003 samples, we again inspected ratios of pigment absorbance at 663 nm before and after acidification (using the methods described in APHA 1993). This ratio can range between 1 and 1.7, with 1.7 representing algae in excellent physiological condition with little

Table 1. Abundances of macroinvertebrates (n/1.35 m^2) and total chlorophyll-a levels at sites in the upper Derwent River upstream and downstream of the WWTP discharge point (15 surber and algal samples per site). U/S, D/S = upstream, downstream sites.

			Pre-discharge						Post-discharge															
Macroinvertebrate abundances			**U/S**	**D/S**	**U/S**	**D/S**	**U/S**	**D/S**	**U/S**	**D/S**	**U/S**	**D/S**	**U/S**	**D/S**	**U/S**	**D/S**	**U/S**	**D/S**	**U/S**	**D/S**	**U/S**	**D/S**	**U/S**	**D/S**
(n/1.35m^2)		Date :	Jan-91	Jan-91	Feb-91	Feb-91	May-91	May-91	Dec-92	Dec-92	Feb-93	Feb-93	May-93	May-93	Jul-93	Jul-93	Feb-94	Feb-94	May-94	May-94	Apr-96	Apr-96	Mar-99	Mar-99
Class	**Order**	**Family**																						
Nematoda	Nematoda		5	5	1	0	0	0	0	1	0	0	0	0	0	0	10	0	0	10	10	0	2	0
Mollusca	Bivalvia	Sphaeriidae	280	260	250	530	870	540	5	30	60	30	250	10	70	30	300	220	640	50	4960	2020	5	57
	Gastropoda	Hydrobiidae	10	0	10	0	30	0	0	0	0	10	10	0	0	0	20	0	0	0	100	80	0	1
		Planorbidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Oligochaeta		690	760	820	820	1640	1370	200	200	330	120	130	350	120	160	380	350	830	860	660	1220	26	45
Hydracarina	Arachnidae	Hydracarina	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	0	0	10	50	0	1
Crustacea	Amphipoda	Paramelitidae	0	10	0	0	0	0	0	10	0	0	0	0	0	0	0	0	0	0	0	0	0	1
	Ostracoda		0	0	0	0	10	20	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
	Isopoda	Phreatoicidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	0	0	0	0	0	0
	Plecoptera	Eusthenidae	20	0	10	10	10	0	0	0	0	0	0	0	10	0	0	0	0	0	0	0	0	0
		Gripopterygidae	0	0	0	10	80	60	0	0	0	0	30	80	190	150	0	0	290	350	110	20	1	4
		Notonemouridae	120	30	0	20	10	0	90	190	0	10	0	0	0	10	30	30	10	0	0	0	0	0
	Ephemeroptera	Leptophlebiidae	860	980	1300	950	1160	1110	180	230	500	360	240	360	310	240	570	350	610	570	1260	1300	134	55
		Baetidae	10	0	60	20	40	20	0	0	0	0	10	0	0	0	10	0	30	40	70	80	0	8
Hemiptera	Hemiptera	Corixidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	10	0	0	0	0
Diptera		Chironomidae	1370	390	900	300	310	240	50	60	360	220	70	30	320	440	210	190	730	270	7380	1060	24	9
		Simuliidae	90	40	40	30	70	40	20	20	130	70	20	60	0	0	60	50	20	20	30	0	2	9
		Tipulidae	0	10	30	0	20	0	0	20	0	0	20	0	0	0	0	0	10	0	50	0	0	0
		Athericidae	20	10	0	20	0	10	0	0	10	0	10	0	10	0	10	0	10	10	0	0	0	0
		Ceratopogonidae	0	0	0	0	0	10	0	0	0	0	0	0	0	0	0	0	0	10	10	10	0	0
		Empididae	0	0	0	0	0	0	0	0	0	10	0	0	0	0	0	0	0	0	0	20	0	0
		Unid. pupae	230	40	130	10	150	30	0	10	30	50	20	40	0	20	30	40	70	60	40	50	19	10
		Calocidae/Helicophidae	40	0	40	40	20	20	0	0	10	0	0	10	10	0	0	10	0	0	0	10	0	0
		Conoesucidae	0	10	10	0	0	10	0	10	10	0	10	0	0	0	0	0	10	0	30	30	1	3
		Ecnomidae	0	20	30	0	0	0	0	0	0	50	10	0	10	30	40	40	30	60	80	140	13	2
		Glossosomatidae	0	0	0	0	0	0	10	0	0	0	0	0	0	0	0	0	0	0	30	0	0	0
		Helicopsychidae	10	0	0	0	10	0	0	0	0	0	0	0	0	0	0	0	10	0	0	0	0	4
		Hydrobiosidae	50	50	100	40	90	40	10	20	50	30	50	60	40	20	10	20	30	60	200	30	2	9
		Hydropsychidae	0	0	0	0	0	10	0	0	0	0	0	0	0	0	0	0	30	30	0	0	1	3
		Hydroptilidae	30	20	20	30	20	50	0	0	0	0	0	0	0	10	0	0	0	10	160	190	0	0
		Leptoceridae	410	310	390	710	460	490	20	20	290	130	180	170	190	160	70	140	410	150	1230	1030	121	57
		Philorheithridae	0	20	40	10	50	10	0	10	60	10	40	20	10	10	10	10	40	10	0	0	0	0
		Polycentropodidae	0	10	0	30	0	40	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
		Unid. pupae	0	0	0	0	0	10	0	0	0	0	0	0	0	0	0	10	0	0	0	0	0	0
	Coleoptera	Adult Elmidae	10	0	0	30	10	30	10	10	20	0	10	0	0	10	30	40	70	30	0	30	8	4
		Larvae Elmidae	20	50	50	90	150	150	30	30	40	10	70	20	10	60	130	70	150	80	200	220	1	15
		Scirtidae	0	0	10	0	10	10	10	0	0	0	20	20	20	30	0	0	0	0	10	10	0	0
		Psephenidae	40	20	40	10	10	20	0	20	0	0	30	40	0	30	70	0	40	20	100	10	5	1
		N taxa	**19**	**19**	**20**	**20**	**23**	**24**	**11**	**16**	**14**	**14**	**20**	**14**	**14**	**16**	**18**	**17**	**21**	**21**	**22**	**21**	**16**	**21**
		Total abundance	**4315**	**3045**	**4281**	**3710**	**5230**	**4340**	**635**	**891**	**1900**	**1110**	**1230**	**1270**	**1320**	**1410**	**1990**	**1590**	**4070**	**2710**	**16730**	**7610**	**365**	**300**
		Bray Curtis Similarity	**0.75**		**0.74**		**0.87**		**0.82**		**0.69**		**0.66**		**0.84**		**0.80**		**0.77**		**0.58**		**0.57**	
Chlorophyll A (mg/m2)		Mean (n = 15)	**13.13**	**5.37**	**16.46**	**7.33**	**25.46**	**27.58**	**14.03**	**18.80**			**30.30**	**20.60**	**59.10**	**44.40**	**5.56**	**4.34**	**26.40**	**15.60**	**8.57**	**6.99**	**3.30**	**5.19**
		Standard deviation	13.58	3.65	12.29	2.06	12.20	11.52	7.94	9.14			12.00	7.60	23.80	16.90	1.83	1.50	10.50	4.00	5.12	2.93	2.99	2.74

pheophytin a pigment (a decay product of chlorophyll-a), and 1.0 representing pure pheophytin a (and hence moribund algae). The ratios for the upstream and downstream sites respectively were 1.46 and 1.55 (with standard deviations of 0.08 and 0.l23 respectively). These values are typical of Tasmanian upland stream algae (Davies unpub. Data), and indicate a healthy algal community at both sites.

Lake level effects

In the 1997 sampling report (Davies and Cook 1998a), significant differences in abundance had been described between pre- and post-WWTP discharge sampling periods, independent of sampling site position. Thus, both upstream and downstream sites were up to 75% lower in abundance post-discharge (1996-97) than prior to discharge commencement (1995). Similar declines in the number of taxa were observed, with a loss of some 4 - 5 taxa in the 1996-97 post-discharge sampling period.

These effects were attributed to the effect of lake inundation. Both sites were continuously inundated by Lake King William which was held at or near FSL for the period May 1995 to December 1996. Inundation causes significant decline in current velocity and increase in depth at both sites, changing the nature of the benthic habitat for invertebrates. Particular taxa were observed to decline as a result - notably eusthenid stoneflies, baetid mayflies, hydroptilid and calocid caddisflies and all bivalves (see Table 1).

The two sites were again inundated by high lake levels for several months prior to sampling in 1998, but not in 2000 or 2003, when lake levels were consistently low (due to prolonged rainfall deficits in central Tasmania).

The overall trend in all sites therefore (Figure 1) is a major decrease in abundance and diversity between May 1995 and December 1996, with some subsequent recovery in abundance to July 1997. This was followed by a further decrease in abundance at both sites in February and May 1998, a substantial increase at both sites in April 2000 followed by a decrease in March 2003. None of these changes can be attributed to the effects of the WWTP discharge.

Despite this effect, the plot of Bray Curtis Similarities (Figure 2) indicates that the upstream and downstream sites maintained highly similar invertebrate communities at all times (mostly > 70%).

WWTP effects

Figure 1 shows that, while there are differences between the upstream and downstream sites in abundance and number of taxa, there is no consistent change in the difference that can be associated during the 'with-discharge' period up to March 2003. The two-factor ANOVA's again confirmed that there was a significant effect of time (pre- vs. post-, $p < 0.0005$) but not of site nor of the site x time interaction (both $p > 0.9$) for these two variables. This indicates that there was no statistically significant change in the difference between upstream and downstream sites with change from pre- to post- discharge period up to and including March 2003.

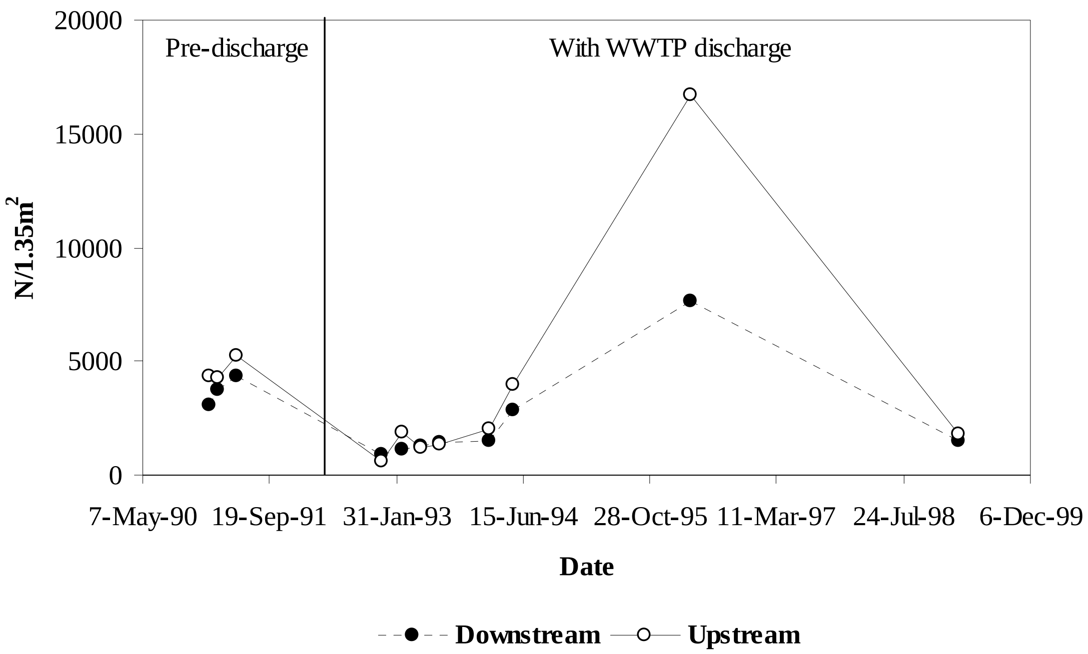
The same result was obtained for the chlorophyll-a data for benthic algae in the Derwent (see Figure 3). There were also no significant difference in March 2003 between the upstream and downstream sites in the index of algal condition (pigment absorbance ratio) indicating that the WWTP was having no significant affect on algal physiological condition.

Examination of the 1997 sample data (Figure 2) had shown that, on two sampling occasions, the Bray Curtis Similarity values fell below 70% - 2/97 and 5/97 (Davies and Cook 1998a). On both of these occasions, lake levels had fallen to the point where the fluvial nature of the sites was re-established. It was suggested that a minor effect of WWTP discharge could have occurred during the summer-early winter period. Recovery in similarity in July 1997 was accompanied by large increases in river discharge and winter temperatures. Similarities were higher again in 1998. Similarity values in April 2000 and March 2003, were lower than previously. However, on inspection of the data, it is apparent that this is due to a higher abundance of chironomids at the upstream site in April 2000 and a generally low abundance at both sites in 2003. There was no evidence of any effect of WWTP discharge on macroinvertebrate community composition in either of these years.

An MDS ordination was constructed (Figure 4) for all invertebrate samples collected to date. All pre-discharge samples were highly similar, while all post-discharge samples were more and unpredictably dissimilar to each other. Samples collected in 12/96 were very different from all 1995 samples, and subsequent samples in 1997 had shown signs of a recovery in community composition and abundance to that in 1995. The suggestion by Davies and Cook (1998a) that this was an effect associated with ‘recovery’ from site inundation was supported by:

- the repetition of this ‘inundation disturbance’ - recovery trend for the 1998 samples; and

- increasing abundance and diversity in the 2000 samples (Figures 1 and 4).



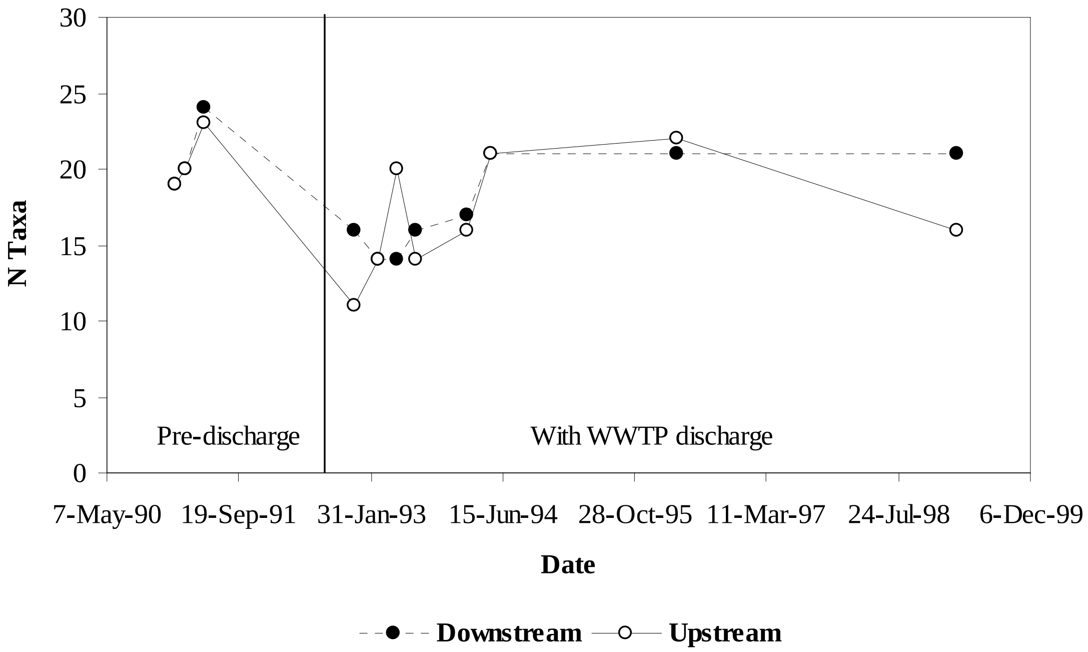


Figure 1. Total macroinvertebrate abundance and number of taxa vs. sampling occasion for the upper Derwent River. Points represent samples from 1995 to 2003 for upstream and downstream sites. Trends are similar for upstream and downstream sites both pre- and with-WWTP discharge, indicating little influence of the discharge .

Sample sin 2003 were not dissimilar from each other, and fell within the range of the other sample pairs. Again, there were no indications of any differences related to the WWTP discharge.

Multivariate analysis of variance (using ANOSIM in the PRIMER software package) again failed to find any significant differences in Bray Curtis similarity between sites ($p > 0.9$), supporting the conclusion that WWTP discharge is having no significant impact on macroinvertebrate community composition.

Once again, no individual taxa could be identified in 2003 that showed differences between upstream and downstream sites that could be associated with a summer-autumn effect of the WWTP discharge.

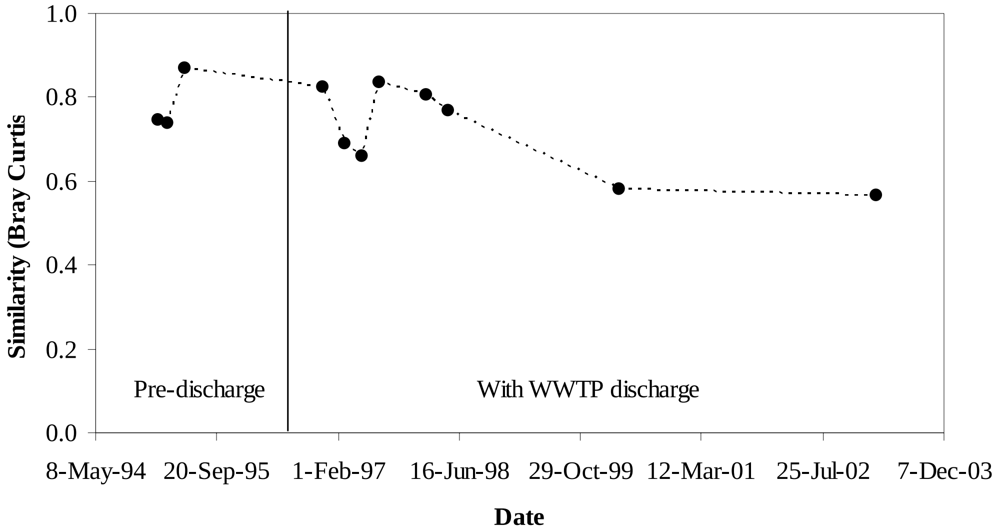


Figure 2. Bray Curtis Similarity of macroinvertebrate communities between upstream and downstream sites in the upper Derwent River for each sampling occasion. High values indicate high similarity, with 1 (100%) being identical, and 0 being completely dissimilar (no taxa in common). Pre- and with-discharge periods indicated. There is little variation in similarity, with generally high levels indicating little compositional difference between the upstream and downstream sites.

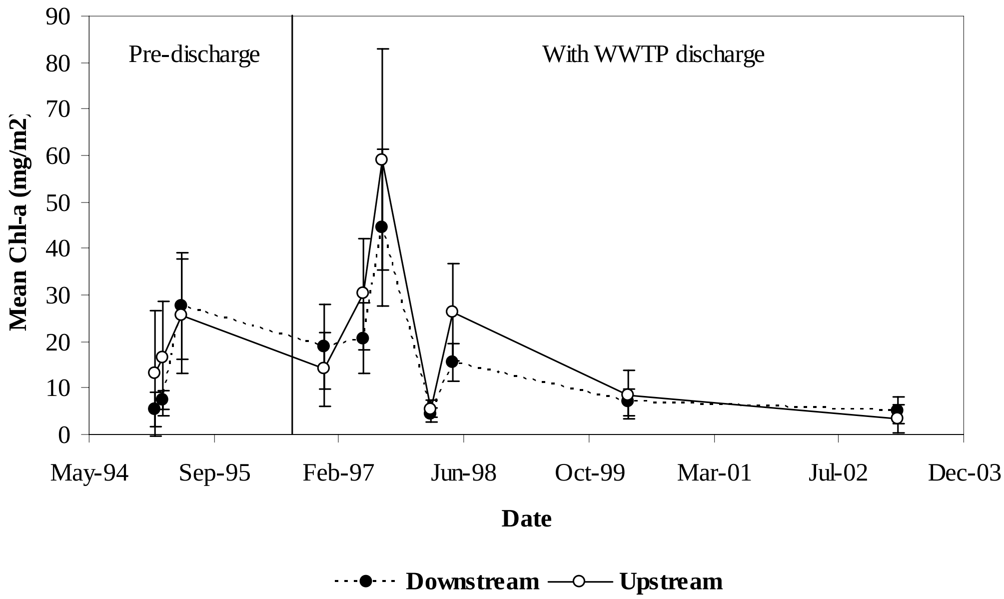


Figure 3. Chlorophyll-a levels for sites upstream and downstream of the WWTP discharge outfall in the upper Derwent River. Standard deviation bars shown. Trends in chlorophyll-a are the same for upstream and downstream sites both pre and post WWTP discharge commencing. Chlorophyll levels were at their highest following prolonged lake inundation (autumn-winter 1997), and were low in 2003.

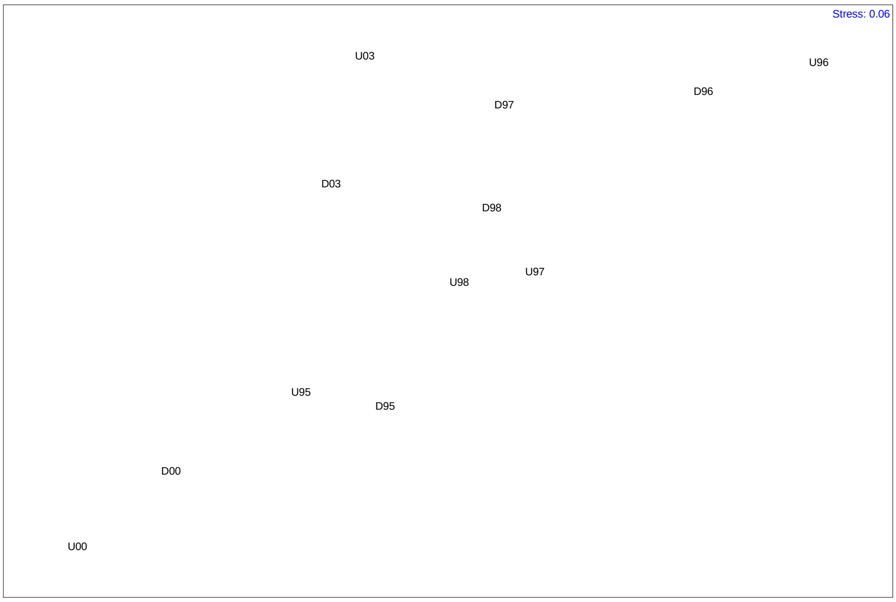


Figure 4. Ordination of macroinvertebrate samples collected in the upper Derwent River at sites upstream and downstream of the WWTP discharge on one occasion in summer-autumn in each year of sampling (Letters indicate site – Downstream or Upstream – and numbers indicate year). The distances between sample pairs (D and U in the same year) are generally similar and do not form a pattern consistent with any impact of the WWTP discharge.

Summary

In summary:

1. Overall abundance of macroinvertebrate was low in 2003 at both sites. However, as in previous years, there were no statistically significant or ecologically significant changes in total abundance or number of taxa (at family level) of macroinvertebrates, nor in algal biomass (as chlorophyll-a) associated with the WWTP discharge, as detected by comparison of differences between the upstream and downstream site values in 2003 and across all years pre- and post- commencement of discharge. There were also no significant differences in physiological condition of benthic algae associated with WWTP discharge.

2. There had been a major effect of inundation from Lake King William on the invertebrate communities at both sampling sites in 1996-97, associated with declines in abundance and diversity. This effect was not observed in 1998, 2000 or 2003. This was probably due to the shorter period of inundation preceding sampling in February 1998, and the prolonged absence of inundation prior to sampling in 2000 and in 2003.

3. Overall, the upper Derwent River continues to sustain a diverse community of macroinvertebrates, and a low abundance of stream algae. The macroinvertebrate community is dominated by taxa (families) which are sensitive to poor water quality. The biota is therefore consistent with this river being unpolluted.

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